EMERGING ISSUES FOR THE RESTORATION OF TIDAL MARSH ECOSYSTEMS IN THE CONTEXT OF PREDICTED CLIMATE CHANGE

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Abstract

There is currently a large regional effort to restore tidal marsh ecosystems in the San Francisco Bay-Delta Estuary involving the commitment of hundreds of millions of dollars and broad landscape-scale habitat manipulations. Although climate change has been on the horizon for many years, recent developments suggest that it must be taken seriously as a factor to be considered in future planning for marsh restoration efforts. Tidal marshes are vulnerable to changes in salinity and inundation rates, both of which will be affected by climate change. Restoration sites may be particularly vulnerable given unpredictable sediment inputs and newly established vegetation. Predicted shifts in snowmelt and altered runoff will change estuarine salinity patterns and could have large-scale impacts on marsh dominance, especially for freshwater marshes. Even relatively small salinity changes could lead to shifts in dominant species, with freshwater marshes being replaced by brackish marshes and brackish marshes converted to salt marsh communities. This will cause a reduction in overall estuarine plant diversity and productivity, with possible reverberations for the estuarine food web. Based on monitoring data from San Francisco Bay marshes, we predict that salinity will have a more immediate impact on tidal marsh vegetation than sea-level rise. However, sea-level rise poses a potentially greater long-term threat, depending on its rate, because the effects of inundation and a more persistent salinity regime could cause widespread marsh loss. If ice sheets in Antarctica and Greenland begin melting at rapid rates, inundation impacts could be catastrophic for coastal marshes. Given the magnitude of these potential changes, we urge the restoration and conservation management community to integrate these contingencies into adaptive management process and to join with the broader community in forging more flexible governance institutions that can respond effectively to large-scale uncertainties and trajectories as they unfold.

Key Words: brackish marsh, climate change, freshwater tidal marsh, inundation, salinity, salt marsh, *Sarcocornia pacifica*, sea-level rise, *Spartina foliosa*.

Growing evidence suggests climate change will have an impact on virtually all ecosystems (Walther et al. 2002). While climate change models have generated a wide range of predictions, a consensus indicates a few critical shifts could affect specific ecosystems, particularly tidal marshes. First, the rise in average global temperatures will influence the timing and degree of snowfall and ice melt, shifting temporal runoff patterns in watersheds dominated by mountain snowpack. Second, rates of sea-level rise are almost certain to increase over the next several decades. Sea-level rise will push sea water farther up into estuaries, increasing salinities in tidal ecosystems. Tidal marshes are likely to be particularly vulnerable to climate change impacts through these shifts in salinity and inundation patterns. For a region like the San Francisco Bay-Delta estuary (hereafter, SF Estuary), less snow and earlier melting of the Sierra snowpack will result in higher floodwaters during winter and early spring but lower flows during late spring and summer. These changes will amplify seasonal and spatial shifts in estuarine salinity patterns, impacting wetland plant establishment, productivity, and reproduction. In addition, tidal marshes will have to accumulate substantial sediment to counteract sea-level rise, or they will be subjected to greater stress associated with longer periods of tidal inundation. The overall influence of climate change, therefore, is a combination of changes in inundation and salinity regimes, with impacts on plant communities and the ecological function of tidal marshes.

Against this backdrop of global climate change and its regional manifestations, large-scale efforts are underway to restore tidal marsh ecosystems, and these initiatives are particularly strong in the SF Estuary. In this paper, we examine tidal marsh restoration in the SF Estuary in the context of climate change and explore the

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potential impacts and vulnerabilities of marsh restoration projects in view of these predicted changes. Our aim is to draw attention to these contextual factors so that they can be more effectively evaluated and mitigated as tidal marsh restoration planning and implementation efforts go forward.

Tidal Marsh restoration

The science of restoration ecology and the practice of ecosystem restoration have grown dramatically over the last decades for a variety of reasons (Falk et al. 2006). Tidal marshes have received considerable restoration attention because many coastal ecosystems have been severely impacted by human modifications and development, with loss of both habitat acreage and function. Within California, for example, over 90% of coastal salt marsh and tidal freshwater marshes have been affected by agriculture, the salt pond industry, airports, and urban development. The recent focus on restoration is evidenced by the number of recent, large-scale projects within the SF Estuary, including the South Bay Salt Pond Restoration Project (over 6000 ha of salt ponds slated for restoration and management), North Bay salt pond restoration, Montezuma wetlands, Sonoma Baylands, Hamilton wetlands, and others. In addition to these large-scale projects, many smaller mitigation projects have occurred throughout the Bay over the last two decades (see www.wetlandtracker.com for mapping of many projects). Most restoration work has focused on salt marsh restoration, with additional large-scale efforts recently undertaken for the SF Estuary's brackish and freshwater tidal marshes through CALFED-funded projects. Similar large- and small-scale efforts exist in southern California, Oregon and Washington.

A major biological focus of marsh restoration efforts has been to maximize plant establishment. In California salt marshes, two dominant species are emphasized, *Spartina foliosa* Trin. (California cordgrass), found along channels and in a narrow band at the low end of the marsh, and *Sarcocornia pacifica* (Standl.) A. J. Scott (pickleweed; formerly *Salicornia virginica* L.), dominant on the marsh plain. In general, the design of restoration projects focuses on elevation as the key factor influencing plant distributions, due to its relation to tidal dynamics and consequently to within-marsh patterns of inundation, anaerobiosis, and salinity. These physical processes appear to be the most important in tidal systems, sorting species by their tolerance to various combinations of these stresses (Mahall and Park 1976a, b, c), although zonation into low and high marsh is an oversimplification of real patterns of vegetation distributions within the marsh (Zedler et al.

1999). Competition and other biological interactions are also important in affecting overall distributions (Pennings and Callaway 1992; Grewell et al. 2007).

Given the focus on elevation as the overall driving force for plant distributions, the approach in early restoration projects was simplistic. Sites were graded to target elevations, and most early restored marshes were designed with a gradually sloping profile from the low marsh through the mid-high marsh. Target elevations across the marsh were based on surveys of plant distributions from nearby natural marshes or from general regional information on elevational distributions. In some cases, propagules (plants, seeds, or cuttings) were introduced to establish appropriate vegetation, in particular for *S. foliosa*; however, in many cases plants were expected to recruit naturally. Restoration projects also were designed without the network of small tidal creeks that are critical to natural marsh dynamics (Zedler et al. 2001; Wallace et al. 2005; Zedler 2005), despite the fact that creeks clearly affect plant distributions (Zedler et al. 1999; Sanderson et al. 2000). Upland transition zones in these restoration projects were often steep and narrow in order to maximize marsh area, a practice that highlights the strong influence of policies that do not count transitional areas or uplands as credit towards mitigation acreage. Maintaining or increasing habitat of rare listed species, such as the Light-footed and California Clapper Rails, has always been one of the key considerations for marsh restoration projects; however, creating this habitat has been difficult (Zedler 1998). Simplistic approaches towards restoration ran into a number of challenges, including, but not limited to incorrect substrate condition (often too coarse, with little organic matter and low nutrient concentrations, especially when using dredged material to build elevations); slow recruitment of native vegetation; invasive plants (in particular *Spartina alterniflora* Loisel. and its hybrids with *S. foliosa* in SF Estuary); unpredictable hydrological complications; improper elevations; and erosion from wind-driven waves across flooded sites (Zedler and Callaway 2000; Williams and Faber 2001; Callaway 2005).

As a response to some of these challenges, there have been shifts in recent restoration designs for tidal marshes. For example many restoration sites are now established at elevations slightly below target elevations for plant recruitment to allow for natural sediment accumulation throughout the rooting zone of marsh vegetation and to stimulate development of tidal creeks (Williams and Orr 2002). This approach has been particularly popular in the SF Estuary where high concentrations of suspended sediment ensure relatively rapid and consistent rates of

sediment accumulation. In Tijuana Estuary, substantial efforts have been made to incorporate experimentation into restoration projects in order to improve restoration design and improve understanding of controls on marsh functioning (Zedler 2001). The Tidal Linkage project focused on the link between plant species diversity and ecosystem functions in restored salt marshes (Zedler et al. 2001; Callaway et al. 2003; Sullivan et al. In Press), while Friendship Marsh, an 8-ha project in the south arm of Tijuana Estuary, has evaluated the importance of tidal creek networks (Wallace et al. 2005). As our understanding of wetlands increased and the significance of linkages to terrestrial and aquatic ecosystems became clearer, the goal of tidal marsh restoration has evolved from the more narrowly focused goals of plant establishment and endangered species towards the maintenance and expansion of community structure, food web dynamics, and ecosystem function. This trend is especially evident in large-scale projects that are not constrained by mitigation issues.

Recently restored tidal marshes are likely to be particularly vulnerable to impacts related to climate change. While shifts in vegetation communities in older marshes will be buffered by the existence of well-established vegetation, recently restored sites lack dense vegetation cover, and new sites have no vegetation at all. Recruitment patterns will determine dominant vegetation in newly restored sites, and recruiting plants are likely to be more vulnerable to extreme events. Therefore, climate-induced shifts in salinity and/or inundation regimes are more likely to cause rapid vegetation shifts at newly restored sites. Reinforcing potential shifts in vegetation establishment, the lack of vegetation at newly restored sites increases surface evaporation and can increase surface soil salinity to the point of salt crust formation in poorly flushed areas of restored sites, leading to even higher salinity stresses (Zedler et al. 2003).

Compared to natural marshes, restored marshes are at a disadvantage under a tidal regime of higher sea level and longer inundation periods. Not only will restored marshes have to keep pace with increased rates of sea-level rise, they also will have to accumulate additional sediment in order to build elevation. Newly restored sites must increase elevation to a point where vegetation is able to colonize the restoration site. Older restored sites may have vegetation established but often need to increase elevation to get to typical marsh plain elevations of mean higher high water (MHHW). Excessive inundation will be especially problematic at highly subsided sites, which are quite common in SF Estuary. Sites that are designed at lower initial elevations with the expectation that they will build up sediment over time also could be particularly vulnerable.

CLIMATE CHANGE IMPACTS

As alluded to above, the impacts of climate change on tidal marsh restoration in the SF Estuary will involve primarily changes in salinity and inundation regimes. In the following section, we develop more detailed insight into the basis for this concern.

Causes of Salinity Changes

As climate change progresses, estuarine salinities (and soil salinities in adjacent tidal marshes) will be affected by shifts in three primary factors: (1) total regional precipitation, (2) seasonal timing of precipitation and runoff patterns (in particular shifts in the amount of snow vs. rainfall and shifts in snowmelt periods), and (3) increases in sea level. Increases in evapotranspiration also have the potential to increase soil salinity, although this is difficult to quantify. There has been substantial debate on what type of changes may occur in terms of total regional precipitation within the state. Dettinger (2005) reviewed a number of the various models and scenarios. While the perception has been that precipitation could change much more than temperature based on model projections, Dettinger (2005) found that projected temperatures spread more widely than projected precipitation in relative terms.

Despite the lack of agreement over what might occur in terms of total precipitation, there is a general consensus that warmer temperatures associated with climate change will lead to less snowfall, more rain, and earlier snowmelt throughout California (Gleick 1987; Gleick and Chalecki 1999; Knowles and Cayan 2002, 2004; Dettinger et al. 2004; Dettinger 2005). In combination, these factors will lead to much earlier runoff within California watersheds, regardless of any changes in total precipitation within the state or region. Stewart et al. (2005) have documented shifts of 1–4 wks in annual spring flows in areas of the western United States. In addition, Stewart et al. (2004) predict substantial shifts in the timing of snowmelt, with a shift of 30–40+ days toward earlier snowmelt in much of the western United States by 2100. This shift will be due primarily to increases in springtime temperatures rather than any predicted shift in regional precipitation patterns. Dettinger (2005) found that predictions of extremely wet conditions in California are extreme outliers based on current projections for climate change. Future scenarios that result in the warmest predictions indicate slightly drier conditions, while those that predict the least warming indicated slightly wetter conditions.

Malamud-Roam and Ingram (2004) and Malamud-Roam et al. (2006) examined stable isotope

data and pollen records from the SF Estuary marsh sediments during the late Holocene and correlated their findings to broader temperature and precipitation patterns in western North America. They found three major periods of higher-than-average salinity in marshes of the SF Estuary that correlated to reductions in fresh water flows during prolonged droughts. Periods of higher-than-average salinity occurred between 1600–1300 cal yr B.P., 1000–800 cal yr B.P., 300–200 cal yr B.P., and from A.D. 1950 to present (Malamud-Roam and Ingram 2004). According to Byrne et al. (2001), the recent increase in Bay salinities is likely caused by damming and diversions of water. Critically, the climate change predictions of shifts in freshwater inflows to the bay and associated salinity changes do not reflect any future increases in human water use, despite the fact that there is near certainty that there will be large increases in California population and water demands. Future increases in human water use are highly likely to reduce any increases in winter/spring inflows and further exacerbate summer reductions in flow.

In addition to salinity shifts due to changes in precipitation and runoff, there will be increases in estuarine salinities due to higher sea level. As sea level rises, it will push more saline water farther into estuarine systems. Uncles (2003) used a hydrological model to evaluate the potential effect of sea-level rise alone (without any change in freshwater inputs) on salinities within SF Estuary. The model predicts salinity shifts based on the location of the 2 ppt bottom isohaline within SF Estuary, known as "X2." X2 is an indicator of the location of the interface between incoming fresh water from the Sacramento and San Joaquin Rivers and salt water from the ocean; it shifts seasonally and annually depending on inflow conditions. Uncles (2003) found that a 25-cm increase in sea level would result in a mean shift of X2 by 1 km upstream, with a maximum shift in some years of 3.5 km. A 50-cm increase in sea level resulted in a mean shift of 1.5 km with a maximum shift of 5.0 km, and a 100-cm increase caused a mean shift of 3 km and a maximum shift of 9 km. These model results indicate that substantial shifts in salinity could occur year round due to sea-level rise, and these changes would be additive to any effects due to shifts in watershed runoff.

Salinity Effects on Vegetation

The changes in salinity due to precipitation, runoff, and sea-level rise will have variable effects on vegetation. The lower salinities that are predicted in the winter and early spring will affect plant recruitment in restored marshes. Larger pulses of winter fresh water could increase recruitment for many species, as most salt and brackish marsh vegetation responds positively to freshwater pulses and reductions in salinity (Ungar 1978, 1991; Noe and Zedler 2001a). Allison (1992) found that increases in late spring rainfall led to increased diversity in salt marshes adjacent to Bolinas Lagoon. Although lower winter and early spring salinities may increase recruitment within salt and brackish marshes, it is not clear how these newly recruited plants might fare with increased salinities in summer and fall. They may not be able to reproduce or survive higher salinities during the growing season, and the influence of more extreme inter-annual variability that is likely with climate change is also very difficult to evaluate.

Higher salinities in the summer and fall will lead to greater stresses on vegetation, leading to reduced productivity and potential mortality (Callaway and Sabraw 1994; Noe and Zedler 2001a, b). In addition, soil salinities within tidal marshes are higher than water salinities within an estuary due to salt accumulation through evapotranspiration; while tidal flushing removes some of the salt, salt accumulates in the soil. This increase in soil salinity will be greater at higher elevations across the marsh, as higher areas are flushed less regularly by the tides and are exposed to surface evaporation for longer periods. As overall salinity within the Bay increases and more salts accumulate in tidal marsh soils, larger pulses of freshwater of greater duration will be required to reduce soil salinities in the marsh and promote germination and recruitment.

Data from a severe drought in 1976–77 indicate that salinity shifts can lead to large-scale changes in plant communities within SF Estuary. Atwater et al. (1979) documented a decrease in both the abundance and height of bulrush species (*Schoenoplectus americanus* (Pers) Volkart ex Schinz & R. Keller, *Schoenoplectus californicus* (C. A. Mey.) Soják, *Schoenoplectus acutus* (Mohl. ex Bigelow) Á. Löve & D. Löve, and *Bolboschoenus maritimus* (L.) Palla [names reflect recent changes in nomenclature in the genus *Scirpus*]) near Carquinez Strait during the drought with the subsequent invasion of these areas by *S. pacifica*. Salinities within the Bay near the eastern end of Carquinez Straits increased from below 10 ppt to 15–20 ppt during this period. Collins and Foin (1992) noted the spread of *S. foliosa* upstream in the north Bay during drought periods, and Malamud-Roam et al. (2006) found similar patterns of vegetation shifts in their synthesis of data from cores over a scale of hundreds to thousands of years. In other regions, similar effects from salinity shifts associated with droughts have been found. In southern California, a severe drought led to increased mortality of *S. foliosa* and long-lasting shifts in the community composition (Zedler et al. 1986). A drought in 2000 led to shifts of up to 5–10 km in

dominant tidal marsh plant communities in Louisiana. Salinities were elevated 3–10 ppt throughout the growing period in Barataria Basin (625,000 ha study area) with large-scale changes in marsh distribution from 1997 to 2000. This included an increase of 8900 ha of salt marsh and 15,900 ha of brackish marsh, and loss of 15,600 ha of fresh marsh and 9200 ha of intermediate marsh (Visser et al. 2002). Shifts were greatest in the marshes that were near large bodies of open water. Measurements from a low salinity area near Lake Pontchartrain, Louisiana during this same drought showed that mean annual salinities ranged from 0–3 ppt over a 50 yr period but were 6.5 ppt in 2000, and higher salinities could affect plant productivity and marsh stability at these sites (Thomson et al. 2001). Greenhouse evaluations of salt pulses with Gulf Coast species have shown that one to three month exposure of elevated salinities ranging from approximately 6 to 12 ppt can lead to shifts in species dominance of oligohaline (0.5 to 5 ppt) marsh species (*Sagittaria lancifolia* L., *Eleocharis palustris* (L.) Roemer & J. A. Schultes, *S. americanus*, and *Panicum hemitomon* J. A. Schultes) (Howard and Mendelssohn 1999, 2000).

Given these historic data and short-term greenhouse effects, droughts or other extreme events associated with climate change could have large-scale effects on marsh vegetation. Droughts are not a direct analog to predicted changes in San Francisco Bay salinities since predictions call for lower late spring/summer runoff but higher winter/early spring runoff; however, increased inter-annual variability in precipitation is likely with climate change (e.g., enhanced El Niño-Southern Oscillation [ENSO] events). The effects of salinity increases associated with pulses of salt water intrusion are likely to be greatest on freshwater vegetation sensitive to small shifts at the low end of the salinity spectrum. This would indicate that any salinity impacts from climate change are likely to be observable first in freshwater marshes in the western delta that are bordered by brackish conditions.

Implications of Salinity Changes for Marsh Restoration

With respect to restored marshes, impacts from salinity shifts or pulses will be greater on newly recruiting plants than on well-established vegetation. Deegan et al. (2005) evaluated shifts in both salinity and inundation regimes on both mature plants and seedlings for two brackish marsh species in Ireland. Mature plants survived but had reduced growth and reproduced at high salinities (10 ppt), while all seedlings died after one week at this salinity level (Deegan et al. 2005). Higher salinities also reduced emergence of most oligohaline species that were tested from

Louisiana seed banks (including *S. lancifolia*, *Polygonum punctatum* Elliot, *Eleocharis parvula* (Roemer & J. A. Schultes) Link ex Bluff, Nees & Schauer, and *Cyperus odoratus* L.) with few species establishing at salinities above 4 ppt (Baldwin et al. 1996). While general predictions indicate the likelihood of greater freshwater inflows to estuaries and reductions in water salinity in winter and spring during recruitment periods, there is also the potential for shifts in extreme conditions associated with climate change. This could cause increased droughts or extremely wet years with very unpredictable effects on long-term trends in vegetation and recruitment.

Within a marsh, salinity impacts are likely to be greatest in the upper marsh due to greater accumulation of salts through evapotranspiration and infrequent flushing, especially during summer months. An indication of the increased effect of salinity in the upper marsh can be seen in the composition of the high marsh in both salt and brackish marshes in SF Estuary. Whereas the low marsh species are typically different (*S. foliosa* in salt marshes and *Schoenoplectus* spp. in brackish and freshwater marshes), there are many similarities in high marsh species (e.g., *S. pacifica*, *Distichlis spicata* (L.) Greene, *Atriplex triangularis* Willd., *Jaumea carnosa* (Less.) A. Gray, *Triglochin maritima* L., *Grindelia stricta* DC var. *angustifolia* (A. Gray) M. A. Lane, and *Frankenia salina* (Molina) I. M. Johnst.). High marsh areas in freshwater marshes may be particularly vulnerable to salinity shifts, as well as transitional/upland areas of other tidal marshes. These areas are of particular concern because high marsh areas contain many of the rare and endangered species that are found in California tidal marshes (Baye et al. 2000). Salinity effects have been shown to be of greater importance for vegetation zonation in low latitude marshes on the Atlantic coasts (Pennings et al. 2005), and greater impacts from salinity are likely on the Pacific Coast where Mediterranean-type climate leads to high soil salinities in salt and brackish marshes. In this regard, it could be that marshes in SF Estuary and northern California become more like those in southern California, which are predominately salt marshes with very small, localized areas of brackish and freshwater marsh.

Atwater et al. (1979) first reported that freshwater marshes of the Delta are characterized by greater plant species diversity than the salt marshes of the lower SF Estuary. As part of the Integrated Regional Wetlands Monitoring Program (IRWM, www.irwm.org), we surveyed plant distributions at six tidal marshes along a salinity gradient from the Petaluma River to the western Delta (Carl's Marsh, Pond 2A, Coon Island, Bull Island, Browns Island and Sherman Lake). Similar to Atwater et al. (1979) we found
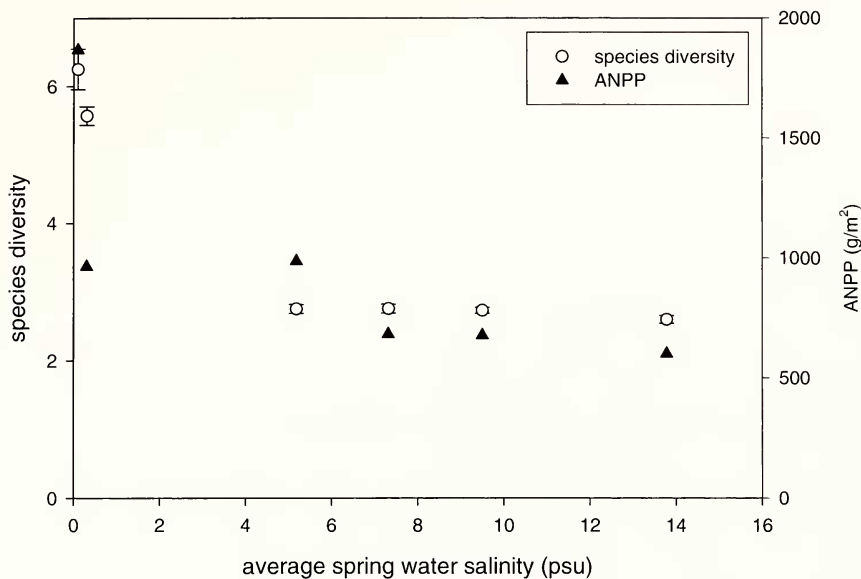
FIG. 1.   Average plant species diversity per 3m-diam. plot and ANPP decrease with increasing salinity in the San Francisco Bay-Delta estuary (error bars = ±1 SE; number of random plots per site range from 151 to 447). Salinity data represent measurements averaged across spring months in 2004 (Wetlands and Water Resources unpublished data). ANPP values were derived from site-specific averages of total standing biomass of individual dominant species that were scaled up to site-level estimates using vegetation maps, and then adjusted by site area to obtain ANPP estimates at the g m^{-2} level. Sites included in order of increasing salinity are: Sherman Lake, Browns Island, Bull Island, Coon Island, Pond 2a, and Carl's Marsh.

a dramatic, non-linear increase in plant species diversity and annual net primary productivity in the fresh and brackish region of the SF Estuary (Fig. 1). Sites that are most saline have relatively low species diversity; however, even sites that are less saline in the upper part of the Napa River are not markedly more diverse. Marshes located farther east in the delta, on the other hand, are substantially more diverse and have greater numbers of locally uncommon and rare species than the four lower SF Estuary sites, including *Lilaeopsis masonii* Mathias & Constance and *Oenothera deltoides* Torr. & Frém. subsp. *howellii* (Munz) Klein. Sanderson et al. (2000) suggested that the low species numbers found at Carl's Marsh, a relatively young restored marsh, are not so much a function of marsh age as its position along the salinity gradient. Despite sampling a total of 1730 0.25-m² quadrats in the Petaluma Marsh, an ancient salt marsh larger than any of our sites, they only encountered a total of 14 species (Sanderson et al. 2000). The greater diversity at freshwater sites underscores the potential ecological importance of freshwater tidal marshes in the upper SF Estuary and their potential vulnerability to salt water intrusion. Given the large number of locally uncommon and rare species in the freshwater tidal marsh ecosystem, as suggested by Lyons et al. (2005) the loss of these marshes could have large consequences for ecosystem function in this region.

As salinity regimes change and plant communities shift, a substantial drop in overall vascular plant productivity is likely within tidal marshes. Increased salinity in a greenhouse experiment reduced both rates of photosynthesis and growth in *S. foliosa* and *Schoenoplectus robustus* (Pursh) M. T. Strong with a small reduction in growth only for *S. pacifica* (Pearcy and Ustin 1984). End-of-year biomass data from a range of the SF Estuary tidal marshes of differing salinity also reflect large increases in productivity with reduced salinity. Mahall and Park (1976a) estimated productivity for *S. foliosa* ranging from 270 to 690 g m^{-2}, and 550 to 960 g m^{-2} for *S. pacifica*. In comparison, Atwater et al. (1979) found end-of-year biomass ranging from 300 to 1700 g/m² for *S. foliosa* and 500 to 1200 g/m² for *S. pacifica*. Similarly, in other estuarine marsh systems, production rates are consistently lower in salt marshes (Odum 1988), likely due to the added stress of high salinities in salt marsh soils. In addition to shifts in overall productivity across our six sites along a salinity gradient (Fig. 1), we found that productivity of *S. pacifica* at three sites across the northern SF Estuary (China Camp, Pond 2a, and Coon Island) was highest in less saline locations within its distribution range (Schile, Callaway, Parker, and Vasey unpublished).

Beyond shifts in plant composition and productivity, there is some evidence that there could be outright loss of some marshes if salinity shifts

are rapid. Marsh loss due to salt water intrusion has been documented in a number of specific cases in Louisiana (Wang 1988; Day et al. 2000). Nyman et al. (1990) identified important implications of tidal marsh soil characteristics for salt water intrusion, with a much greater need for mineral sediment accumulation with increasing salinity. In order to build the same depth of soil, salt marshes required almost twice as much mineral sediment as in brackish marshes and four times that in freshwater marshes (Nyman et al. 1990). Given these requirements, as well as existing differences in soil bulk densities across salinity gradients, the substrate in freshwater marsh may not immediately support a brackish marsh if salinities increase abruptly. Increased salinity stress associated with the 2000 drought was an important factor in the large-scale die-off of salt marshes in Louisiana and Georgia; however, in both cases other factors were also critical, including likely drops in soil pH associated with drying of soils in Louisiana (McKee et al. 2004) and intense herbivory by snails in a variety of locations (Silliman et al. 2005).

Sea-level Rise and the Impacts of Inundation

Recent rates of global sea-level rise have been approximately 1–3 mm yr^{-1} over the last century (IPCC 2001). The recent report from the Intergovernmental Panel on Climate Change (IPCC) documented a global rate of 3.1 mm yr^{-1} from 1993 to 2003 compared to 1.8 mm yr^{-1} from 1961 to 1993, although it is not clear if this increase is due to normal decadal variability or a longer-term trend (IPCC 2007). The increase in sea level is due to thermal expansion of ocean waters and increased inputs of water from melting glaciers and ice sheets. A recent evaluation of tide gauge data indicates that the average rate over the 20th century was 1.7 mm yr^{-1} with an acceleration over that period of 0.013 mm yr^{-2} (Church and White 2006). If this rate of acceleration were to remain constant over the next century, it would result in sea-level rise of 28–34 cm by 2100, with an annual rate of 3–4 mm yr^{-1} by the end of the century (Church and White 2006). The IPCC has intensively evaluated future rates of global sea-level rise, using a range of emissions scenarios, and their predictions indicate a potential increase from 1990 to 2100 of 0.09 to 0.88 m across these scenarios (IPCC 2001; Fig. 2). However, these predictions have not included potential impacts from the melting of large-scale ice sheets in Greenland or Antarctica, which could increase these predictions substantially (see below). Most of the scenario predictions were in the range of 0.3 to 0.5 m over this period. Additional evaluation of model uncertainty shows a similar range of variation based on the use of different atmospheric-ocean

general circulation models (11–77 mm by 2100 for a single scenario). Based on the IPCC results, it appears likely that sea level will increase approximately 15 cm by 2050, with a possible range of 5 to 30 cm (based on the 1990 baseline). Predicted increases diverge by 2100, with overall increases by that time ranging from 0.2 to 0.7 m (Fig. 2). However, recent projections of sea-level rise based on semi-empirical relationships to changes in global mean surface temperature indicate that sea-level rise could be greater than previous IPCC predictions (Rahmstorf 2007). Using the IPCC scenarios of global warming, Rahmstorf (2007) estimated potential increases in sea level ranging from 0.5 to 1.4 m by 2100.

More rapid melting of global ice sheets, primarily in Greenland and the Antarctic could lead to even greater rates of sea-level rise. Flow rates of outlet glaciers associated with the Greenland ice sheets have increased dramatically in the last decade (Rignot and Kanagaratnam 2006). Evaluation of paleoclimate records of melting and sea-level rise also indicate that future rates of both melting and sea-level rise may be much greater than current predictions (Overpeck et al. 2006). Substantial uncertainty remains concerning rates of melting for these ice sheets; however, the potential for large contributions from these sources should be considered. If these ice sheets were to melt completely, the Greenland ice sheet would contribute approximately 7 m to sea level, while the West Antarctica ice sheet would add 8 m. Globally, ice sheets could add 70 m to sea level (Alley et al. 2005). While melting at this scale is unlikely even over many centuries, it possible that melting ice sheets could increase global sea level by a meter or more over the next century (Overpeck and Cole 2006).

Increases in global sea-level rise will lead to increased rates of inundation across all intertidal ecosystems. Within any particular tidal marsh, relative sea-level rise (RSLR) is also affected by local factors that can add to or reduce the effects of global sea-level rise, including subsidence and surface sediment compaction, mineral and organic sediment accumulation, organic matter decomposition, and tectonic activity (Callaway et al. 1996; Nuttle et al. 1997). Under current conditions, most tidal marshes are relatively stable; increases in global sea level or other factors listed above are counterbalanced by sediment accumulation, and the marsh plain maintains elevations approximating mean high water (MHW) to MHHW (Redfield 1972; MacDonald 1977; Callaway et al. 1996). Any major shifts in the relative balance of these factors will lead to changes in elevation, e.g., rapid sediment accumulation from the watershed can lead to supratidal elevations and the conversion of marsh to transitional or upland habitat (Greer and Stow 2003; Callaway and Zedler 2004), while loss of
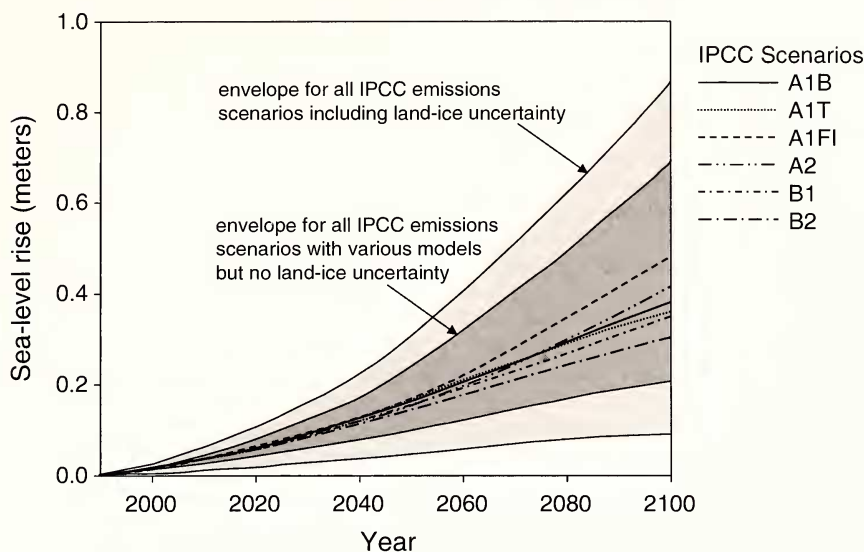
FIG. 2.   Global average sea-level rise from 1990–2100, based on six scenarios from the Special Report on Emission Scenarios (SRES) from the Intergovernmental Panel on Climate Change (IPCC). See IPCC (2001) for details on scenarios. Each line represents average predictions from runs of multiple general circulation models. The region in dark shading represents the range of all general circulation models for all 35 SRES scenarios; the region in light shading represents the range of all models and scenarios including uncertainty in land-ice changes and other factors. Adapted from IPCC (2001).

elevation can lead to habitat shifts (Warren and Niering 1993; Donnelly and Bertness 2001) or loss of marsh (Baumann et al. 1984; DeLaune et al. 1994; Day et al. 1995). Tectonic activity can cause increases or decreases in relative elevation; Reed (1989) documented massive subsidence in Chile which allowed for subsequent development of a tidal marsh over a former upland forest, while uplift associated with tectonic activity has also shifted intertidal plant communities in Alaska (Thilenius 1990). High rates of local subsidence reduce relative elevation, causing increased rates of inundation and potentially marsh loss. This is occurring in a number of deltaic systems including the Mississippi River (Baumann et al. 1984; Day et al. 2000) and the Nile (Stanley and Warne 1993; Stanley and Warne 1998), and in the Chesapeake Bay (Kearney and Stevenson 1991; Ward et al. 1998). The ongoing loss of marshes in areas with high rates of local subsidence can be used as an analogy for potential widespread effects of future increased rates in global sea level, especially as the rates of subsidence in these areas (approximately 1 cm/yr) are at the high end of predictions for global sea level-rise in the next century.

Response of Marsh Vegetation to Inundation

Marsh plants have a number of adaptations that allow them to tolerate some level of inundation. Many species found along creek banks and at lower elevations, in particular *Spartina* spp., *Schoenoplectus* spp., and *Typha* spp., have well-developed aerenchyma, internal tissue with large, connected air spaces that allow for the passage of gases from plant shoots to roots (Armstrong 1979; Kludze and DeLaune 1996; Maricle and Lee 2002). Aerenchyma enables plant roots which are growing in anaerobic conditions to use oxygen from aboveground and may also result in oxygen diffusion into soils adjacent to roots and rhizomes (Howes and Teal 1994). In addition to the structural adaptation of aerenchyma, marsh plants have physiological adaptations that allow them to tolerate inundation, including shifting from aerobic to anaerobic respiration (Mendelssohn et al. 1981; Burdick and Mendelssohn 1990; Maricle et al. 2006). Despite these adaptations, marsh plants still have limits to the level of inundation that they can tolerate; as they are flooded more and more, they have less energy available for growth, and even the most tolerant plants will be subject to stresses that restrict the lower limits of their distribution.

The key questions of interest for the future stability of restored marshes are: (1) how much of an increase in sea-level rise can well-established restored marshes tolerate, and (2) will recently restored marshes be able to develop under increased rates of sea-level rise? In terms of well-established restored marshes, the range of possible outcomes is similar to those for natural marshes. They could (1) accumulate sediment in pace with increases in sea level with little change in overall vegetation; (2) undergo inundation but migrate landward towards higher elevations with local shifts towards more inundation tolerant

plant communities; or (3) be inundated quickly or have no opportunity for migration, resulting in the conversion to unvegetated mudflats. Recently restored sites will either accumulate enough sediment to build in elevation and develop as a vegetated marsh, or they will not accumulate enough sediment to reach threshold elevations for plant establishment and remain as unvegetated mudflats.

If we first consider well-established marshes, accretion and elevation data from a range of natural marshes with varying rates of local sea-level rise (due primarily to different subsidence rates) give insight into possibilities for long-term stability. Most marshes accrete sediment at a rate of 1–5 mm yr^{-1} while some may accumulate up to 10–15 mm yr^{-1} (Stevenson et al. 1986; Reed 1990; Callaway et al. 1996). Most tidal marshes appear to be relatively stable with little shift in vegetation or loss of elevation; however, some may be experiencing slight changes in elevation and resulting shifts in vegetation (Warren and Niering 1993). Donnelly and Bertness (2001) used plant macrofossils in marsh sediment cores to document that *S. alterniflora* has migrated landward in two New England marshes since the late 19th century, paralleling increases in local sea level. Some tidal marshes have much greater sediment deficits compared to marshes maintaining suitable elevations and are being converted to unvegetated mudflats (e.g., Louisiana and Chesapeake Bay, see details below). Stevenson et al. (1986) showed a strong positive correlation between local rates of RSLR and vertical accretion, indicating that marshes respond to increased rates of inundation by accumulating additional sediment. This is to be expected since areas that are lower in elevation and inundated for longer periods should have more potential for sediment accumulation (Pethick 1981; Krone 1987; French 1993; Orr et al. 2003). Availability of mineral sediments, either from local watersheds or reworked estuarine sources, will also affect rates of sediment accretion (Williams and Orr 2002). In addition, organic matter provides much of the structure for marsh sediments, and there is a strong positive relationship between organic matter accumulation rates and vertical rates of sediment accretion (Bricker-Urso et al. 1989; Nyman et al. 1990; Callaway et al. 1997). This implies a positive feedback between marsh productivity and maintenance of elevation and suggests that as marshes are subjected to increased inundation, they could hit a threshold elevation that will lead to greater plant stress and a reduction in organic matter accumulation and vertical accretion (DeLaune et al. 1994; Morris 2006).

Two well-researched examples of tidal marshes that have not been able to keep pace with RSLR include Louisiana coastal marshes associated with the Mississippi River Deltaic Plain (Bau-

mann et al. 1984; Day et al. 2000) and brackish marshes in Nanticoke Estuary within Chesapeake Bay (Kearney and Stevenson 1991; Kearney et al. 1994; Ward et al. 1998). Many measurements of sediment accretion, subsidence, and changes in relative elevation have been completed in Louisiana using a wide variety of methods, and many of the tidal marshes in the Mississippi Deltaic Plain are not keeping pace with rates of local sea-level rise (DeLaune et al. 1978; DeLaune et al. 1983; Hatton et al. 1983; Baumann et al. 1984; Nyman et al. 1990; Boesch et al. 1994; Day et al. 2000; Turner 2001), while Gulf Coast marshes in more stable areas outside the delta, such as the Chenier Plain, and the Texas and Mississippi coastlines are keeping pace (Callaway et al. 1997). The Louisiana marshes appear to be able to withstand RSLR of up to 10 or 12 mm yr^{-1}, but at higher rates of RSLR they become submerged. In the Chesapeake Bay case, accretion rates were up to 7.4 mm yr^{-1}; however, local subsidence was very high, and marshes were not able to keep pace with sea-level rise. A number of other coastal marshes have reported marsh loss, although causes may vary. Elkhorn Slough has lost substantial salt marsh area over the last five decades, although exact causes are unclear, and loss here is more likely due to increased tidal currents and reduced sediment inputs rather than increases in sea-level rise (Van Dyke and Wasson 2005).

In addition to empirical data from natural marshes, modeling studies can provide some insight into what rates of sea-level rise are sustainable for tidal marshes. Morris et al. (2002) evaluated effects of sea-level rise on South Carolina tidal marshes using a model developed to evaluate marsh stability. Model results indicated that with relatively high rates of sediment loading as are found in this area, the marsh would be stable up to a rate of sea-level rise of 12 mm/yr. Rates higher than this led to marsh submergence and loss of vegetation. The modeled marsh had a slightly smaller tidal range (1.4 m mean tidal range) than found in San Francisco Bay. Morris et al. (2002) also found that the most stable area of the marsh was the marsh plain, even though the most productive part of the marsh was the low marsh. Orr et al. (2003) developed a model for SF Estuary marshes using the approach of Krone (1987) and French (1993). Their model tested rates of sea-level rise of 1, 3, 5, 6, and 11 mm/yr and indicated that high elevation marshes would be stable at rates of 6 mm yr^{-1} or below; low marshes were more vulnerable to increased rates, similar to Morris et al. (2002). Shifts in suspended sediment concentrations had a substantial effect on accretion rates and marsh stability. From empirical data, Orr et al. (2003) also concluded that freshwater marshes would be less affected by increases in sea-level rise

and reduced inputs of mineral sediments as they are able to accumulate organic sediments at a higher rate than salt or brackish marshes. In summary, data from natural marshes and simulation models both indicate that inundation rates on the order of 10–15 mm yr^{-1} will lead to marsh loss for well-established marshes, while rates lower than this but greater than current rates could lead to shifts in plant communities.

Implications of Inundation for Restoration

Newly restored tidal marshes are likely to be even more sensitive to increased rates of sea-level rise for a variety of reasons. First, restored marshes need to increase elevation, not just maintain elevation, in order to reach threshold elevations for plant establishment. As outlined above, many current tidal restoration projects are designed at elevations 20–50 cm below target elevations for marsh vegetation (Philip Williams and Associates Ltd. and Faber 2004). While leaving sites at low elevations and allowing natural sediment accumulation to occur has substantial benefits, this approach creates an instant sediment need for restored marshes that could make them more vulnerable to increased rates of sea-level rise, especially because low marsh areas appear to be more vulnerable to sea-level rise than high marsh areas. In addition, some restoration sites may be at even lower initial elevations, due to locally high rates of subsidence rates, organic matter oxidation, and the lack of sediment inputs when sites are behind levees. This is particularly important within south San Francisco Bay, where salt pond restoration sites have subsided anywhere from 50 to 200 cm. Areas in the Sacramento-San Joaquin Delta may be at even lower intertidal elevations (Mount and Twiss 2005).

Second, sparse vegetation (or lack of vegetation) will reduce sediment accumulation rates in restored sites. In tidal marshes, the presence of vegetation slows down tidal currents and promotes sediment accumulation (Temmerman et al. 2005). While the presence of vegetation may not be critical at low mudflat elevations, as mudflats build up to higher elevations, vegetation promotes sediment accumulation (and prevents wind waves and sediment erosion). If vegetation does not establish quickly within restored marshes, this could further limit sediment accretion rates and reduce the ability of restored marshes to keep pace with increased rates of sea-level rise.

Finally, many restored tidal marshes, especially those in urban areas, lack adjacent uplands for possible migration. In cases where there are gradual transitions to adjacent uplands, marshes may migrate inland with increases in sea-level rise (e.g., Warren and Niering 1993; Donnelly and Bertness 2001). Substantial research in the 1980's evaluated opportunities for marsh migration from a policy perspective, and in coastal areas with very shallow slopes and where infrastructure has not been established in adjacent transitional areas, there are opportunities to allow marshes to migrate upslope in the face of higher rates of sea-level rise (Titus 1990, 1991). This approach has been called managed realignment in the U.K. and has been widely promoted there (Pethick 2001, 2002). However, in most cases within California, this type of marsh migration will not be possible because of either naturally sharp topographic breaks between marshes and adjacent uplands or, more commonly, loss and/or degradation of transitional upland areas due to substantial urban, residential, industrial, or agricultural development. Because of their association with mitigation impacts and for a variety of other reasons, restored marshes often lack natural transitional habitats. If restored marshes are surrounded by urban development, dikes or other impacted areas, they will not be able to migrate inland, and increased rates of sea-level rise will lead to marsh loss.

CONCLUSIONS AND RECOMMENDATIONS

Climate change will have substantial effects on tidal marshes in the SF Estuary and across the Pacific Coast. Initial impacts are likely to be caused by salinity changes as a result of altered flow regimes, with even relatively small salinity changes potentially causing shifts in dominant vegetation. Although tidal marsh restoration projects in general will be adversely affected by increasing salinity, tidal freshwater marshes will be the most vulnerable. They are likely to be converted to brackish marshes, with existing brackish marshes converted to salt marsh communities. These shifts will cause a loss in overall tidal marsh plant diversity and a reduction in vascular plant productivity, changes that will cascade into the associated estuarine food webs

While the impacts of inundation associated with sea-level rise may be slower to affect tidal marshes, sea-level rise could ultimately have a much more profound cumulative impact due to the combined effects of excessive inundation and more persistent changes in salinity. Low rates of predicted sea-level rise will cause shifts from marsh-plain to low-marsh vegetation, while rates at the high end of predictions could cause widespread loss of tidal marsh habitats, especially if ice sheets on Greenland and west Antarctica begin melting at a more rapid rate. In addition, the impacts of climate change are not likely to affect tidal marshes uniformly across large scales. Along the Pacific Coast, from Washington to southern California, inundation and anaerobiosis are likely to play a larger role to the north, while increases in salinity, especially in the marsh plain,

TABLE 1.   SUMMARY OF TIDAL MARSH RESPONSES TO POTENTIAL SEA-LEVEL RISE CONDITIONS.

Scenario	Cumulative sea-level rise by 2100 (cm)	Annual rate of sea-level rise (mm/yr)	Marsh dynamics
Historic conditions	10–20	1–2	• restored marshes develop • natural marshes keep pace
IPCC mid-range estimates	30–50	4–6	• restored marshes slower to develop • some shift in vegetation
IPCC high-range estimates	60–90	7–10	• restored marshes unlikely to develop • large-scale shifts in vegetation • likely marsh loss
Recent high estimates	>1 m	>10	• substantial loss of both natural and restored marshes

will constrain marshes in the south. Consequently, restoration ecologists will need to consider factors such as local and regional climate variables, potential shifts in summer rainfall, and how watersheds are likely to change under different climate scenarios.

In terms of management opportunities, water management will be critical for potential salinity shifts; however, opportunities will be constrained by growing human water needs as well as the magnitude of seasonal shifts in runoff. In dealing with increases in future sea level, the greatest opportunity for preserving both natural and restored marshes will be to allow for landward migration of tidal marshes. However, as with water management, there are substantial socioeconomic constraints to dealing with adjacent land use. Purchasing areas adjacent to marshes to permit migration, rather than constructing new marshes directly will be an important component of future marsh restoration and mitigation. This also may require grading and restoration of these adjacent uplands to make the landward sites suitable for future marsh migration. On a broader scale, there is the need to identify opportunities for regional restoration/conservation of sites that will maximize preservation of tidal marsh plant diversity, in particular for tidal freshwater sites that are likely to be impacted by shifts in estuarine salinity. In considering impacts of marsh shifts on animals, the relative importance of various tidal marsh habitats for fish, birds and other target species of interest should be evaluated. Finally, sediment accumulation is a critical issue for marsh stability, and sediment within the estuary sediment should be managed as a valuable resource. Opportunities for building marsh elevation with sediment reuse should be evaluated, as well as other management activities such as small-scale sediment fences which could maximize sediment retention within restored tidal marshes.

Given these challenges and future uncertainties, it is critical that knowledge of potential impacts from current climate change models be incorporated into strategies for the design, implementation, and adaptive management of tidal marsh ecosystems. As emphasized by Millar and Brubaker (2006) and Watson (2004) for the SF Estuary, paleohistoric patterns reveal that the SF Estuary has been extraordinarily dynamic, and it is highly unlikely that the estuary will return to target conditions from the mid 19th century. Rather, we need to adopt a flexible and adaptive approach to tidal marsh restoration in the SF Estuary that will focus more on on-going adjustment to future conditions, opportunities, and trajectories. The SF Estuary is a resource of global importance and a cornerstone to the identity of the Bay-Delta Community. Accordingly, investment in its ecological recovery through marsh restoration is both appropriate and wise. However, to insure that this investment realizes its full potential, the conservation and restoration community must take climate change issues to heart and begin the task of working together in conjunction with the broader society to forge institutions capable of a more nimble response to this challenge.

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